Application of Radioactive Fallout Cesium-137 for Measuring Soil Erosion and Sediment Accumulation Rates and Patterns: A Review

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ABSTRACT

Radioactive fallout ^{137}Cs (cesium-137) deposited across the landscape from atmospheric nuclear tests is strongly absorbed on soil particles limiting its movement by chemical and biological processes. Most ^{137}Cs movement in the environment is by physical processes; therefore, ^{137}Cs is a unique tracer for studying erosion and sedimentation. Cesium-137 loss from a watershed has been shown to correlate strongly with soil loss calculated by the Universal Soil Loss Equation (USLE) or measured from small runoff plates. By measuring spatial patterns of ^{137}Cs in vertical and horizontal planes across the landscape, rates of soil loss or deposition can be measured for different parts of a watershed. Even within landscape units, redistribution of soil can be mapped and erosion or deposition rates for different parts of individual fields measured and mapped. Sediment accumulation rates can be measured by comparing the vertical distribution of ^{137}Cs in sediments with the temporal deposition of fallout ^{137}Cs from the atmosphere to locate sediment horizons. Using these dated sediment horizons, sediment accumulation rates can be measured. Interpretations about the location of these dated horizons must consider particle size of the sediments, reworking of deposited sediments, diffusional movement of ^{137}Cs, and time rates of physical process in the sedimentation process. The ^{137}Cs technique can be used to determine sediment accumulation rates in a wide variety of depositional environments including reservoirs, lakes, wetlands, coastal areas, and floodplains. The bibliography shows that ^{137}Cs has been used widely for studying erosion and sedimentation in many different environments around the world.

SOIL EROSION and deposition of eroded soil particles in fields, floodplains, and water bodies are major environmental concerns around the world. Although soil erosion is a natural process, many human activities have increased rates of soil erosion. Soil erosion reduces soil productivity, scars the landscape, and causes downstream damage. Sediment accumulation in water bodies affects water quality, causes loss of storage capacity, affects biological activity, and reduces recreational potential. Better information is needed on sources, locations, and rates of erosion and on locations, patterns, and rates of deposition of eroded materials. Many qualitative and quantitative techniques have been developed and used to measure patterns and rates of erosion and sediment accumulation. Estimates of these patterns and rates have also been made using empirically and theoretically derived erosion models. However, few methods are capable of providing data on both erosion and sedimentation. Information on the relationship between erosion rates, delivery rates of eroded products, and sediment accumulation rates is needed if erosion and sedimentation is to be understood and monitored on a watershed.

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Radioactive fallout ^{137}Cs has been used as a tracer to provide independent information on erosion and sedimentation rates and patterns. Sedimentation and sediment accumulation in this paper is used to indicate a vertical deposition of sediments (cm yr^{-1}). The purpose of this article is to review the development and application of a technique using radioactive fallout ^{137}Cs to measure rates and patterns of erosion and sedimentation. A bibliography is also provided.

WHY CESIUM-137?

Properties of radioactive ^{137}Cs make it unique as a tracer for studying erosion and sedimentation. There are no natural sources of ^{137}Cs. Cesium-137 is produced during nuclear fission. Thus, its presence in the environment is due to nuclear testing or to releases from nuclear reactors (Wise, 1980; Walling et al., 1986b). Cesium-137 was first released around sites used for early nuclear tests in 1945 (Carter and Moghissi, 1977). Widespread global dispersal of ^{137}Cs to the environment began with high-yield thermonuclear tests in November 1952 (Perkins and Thomas, 1980). In these tests, ^{137}Cs (and other radionuclides) was injected into the stratosphere where it circulates globally (Longmore, 1982). Cesium-137 moves from the stratosphere back to the troposphere and re-entry from the troposphere to the earth's surface (fallout) is strongly related to local precipitation patterns and rates (Davis, 1963; Longmore, 1982).

Temporal and spatial distribution of radioactive fallout on the earth's surface has been extensively measured and documented (Cambray et al., 1983; Larsen, 1985). Based on radioactive fallout measurements by the Health and Safety Laboratory (1972), initial input of ^{137}Cs into the global environment was estimated to be 1952 $\pm$ 2 yr (Robbins et al., 1978) with measurable amounts in soils generally beginning in 1954 (Wise, 1980; Longmore, 1982). Major periods of global deposition of ^{137}Cs fallout in 1958 and 1963/1964 and minor periods in 1971 and 1974 can be related to activity in aboveground nuclear testing (Carter and Moghissi, 1977). Periods of lower fallout rates can be related to moratoriums on testing (1958–1961) and the Test Ban Treaty of 1963. Because of the 1963 Test Ban Treaty, global radioactive fallout rates have decreased steadily except for minor periods in 1971 and 1974 caused by aboveground nuclear testing by nontreaty countries. The 1971 period was more significant in the southern hemisphere than in the northern hemisphere. Cambray et al. (1985) reported fallout deposition rates of ^{137}Cs in the northern hemisphere to be below their limits of detection in 1983 and 1984. Total fallout, as illustrated by ^{90}Sr (Fig. 1), is greater in the northern hemisphere than in the southern hemisphere (Larsen, 1985), because more atmospheric nuclear testing took place in the northern hemisphere. Local events, such as the Chernobyl accident (Volchok and Chieco, 1986), have significant impacts on regional

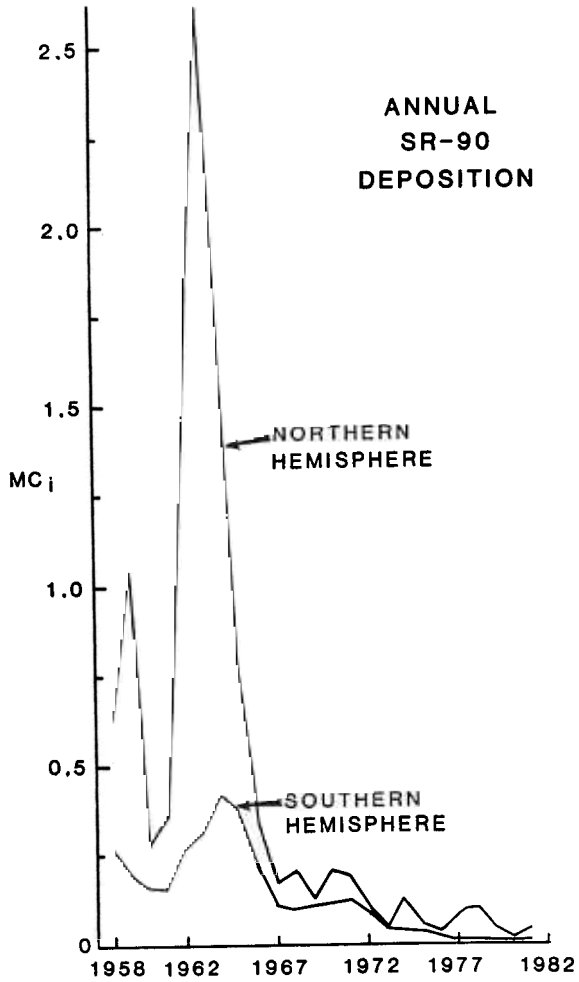

Fig. 1. Annual atmospheric fallout (^{90}Sr) in the northern and southern hemispheres. Cesium-137 is equal to 1.45 time ^{90}Sr at formation.

patterns of fallout radioactivity but have limited impacts on global fallout rates and patterns.

Cesium-137 is strongly adsorbed on clay and organic particles (Tamura and Jacobs, 1960; Schulz et al., 1960; Davis, 1963; Tamura, 1964a,b; Schulz, 1965; Brisbin et al., 1974) and is essentially nonexchangeable (Davis, 1963; Lomenick and Tamura, 1965). Adsorption on soils and sediments is rapid (Eyman and Kevern, 1975) with distribution in undisturbed soil profiles showing an exponential decrease with soil depth (Beck, 1966; Ritchie et al., 1970a, 1972b). Cesium-137 in plowed soils shows a uniform distribution through the plow layer (Cline and Rickard, 1972; Ritchie and McHenry, 1973a). Increased concentrations of competing ions such as Na, K (Coleman et al., 1963; Aston and Duursma, 1973), and H (Agre and Korogodin, 1960; Lerman and Lietzke, 1975) slightly decrease ^{137}Cs adsorption. However, Lomenick and Tamura (1965) removed less than 1% of adsorbed ^{137}Cs on soil and sediment samples using different acids and bases illustrating the strong adsorption between ^{137}Cs and soil particles and the limited mobility of ^{137}Cs by chemical processes.

Cesium-137 emits a strong gamma-ray (662 keV) making its measurement in environmental samples relatively easy and accurate without special chemical preparation or separation (Ritchie and McHenry, 1973b; McCallan et al., 1980). Cesium-137 can be measured using a lithium-drifted germanium semiconductor detector or a thallium-activated sodium iodide crystal and a multichannel analyzer to separate and quantify the 662 keV gamma-ray peak from the gamma-ray spectrum. This specialized equipment is available in radiochemistry and physics laboratories.

With its long half-life (30 yr) and the total deposited fallout especially in the northern hemisphere (Cambray et al., 1983; Larsen, 1985), ^{137}Cs will be detectable in environmental samples for many years.

CESIUM-137 CYCLE ON THE LANDSCAPE

Although dry deposition is important locally around nuclear test sites, on a global scale, ^{137}Cs deposition from the atmosphere to the landscape (Fig. 2) has been found to be strongly related to rainfall with total fallout varying linearly with rainfall within latitudinal zones (Davis, 1963). Transport of ^{137}Cs across the landscape involves transfers between three primary components: vegetation, soil, and water.

Cesium-137 deposited on vegetation may be adsorbed or absorbed. Most adsorbed ^{137}Cs is washed from vegetation and moved to the soil (Davis, 1963; Dahlman et al., 1975). Rogowski and Tamura (1970a) found that 93% of the ^{137}Cs applied to grass washed off during the first year. Absorbed ^{137}Cs is released to soils when vegetation dies and decays. This turnover rate for absorbed ^{137}Cs depends on vegetation type and climate conditions. Uptake of ^{137}Cs by vegetation from soils (Fredrickkson et al., 1958; Davis, 1963; Dahlman et al., 1975) or water (Eyman and Kevern, 1975; Garten et al., 1975) is low. Uptake is far less important than absorption from direct deposition on foliage in determining ^{137}Cs concentration in vegetation (Dahlman et al., 1975). Removal of ^{137}Cs from the landscape with harvested crops is very small (Brown et al., 1981a).

Cesium-137 in soils is due to direct deposition from the atmosphere, wash-off from vegetation, turnover from vegetation, redeposition of eroded soil particles, and deposition from water on floodplains and coastal regions. Loss by biological uptake from soils by vegetation can be considered negligible (Dahlman et al., 1975). Because ^{137}Cs is strongly adsorbed to cation exchange sites (Davis, 1963; Eyman and Kevern, 1975), movement in soils due to chemical or biological processes is limited (Tamura, 1964b; Wise, 1980; Campbell, 1982). Thus, physical processes, such as erosion and tillage, are major causes of redistribution of ^{137}Cs in soils and movement of ^{137}Cs from soils to water.

Cesium-137 in water is from direct deposition on water surfaces and input of ^{137}Cs adsorbed on eroded soil particles. Some fresh fallout ^{137}Cs may move from the landscape during large rainfall events before it becomes adsorbed by soils, but such movement of ^{137}Cs is very small (Brown et al., 1981a). Cesium-137 deposited on water surfaces will be adsorbed by suspended materials in water (Armstrong and Gloyna, 1969; Eyman and Kevern, 1975; Francis and Brinkley,

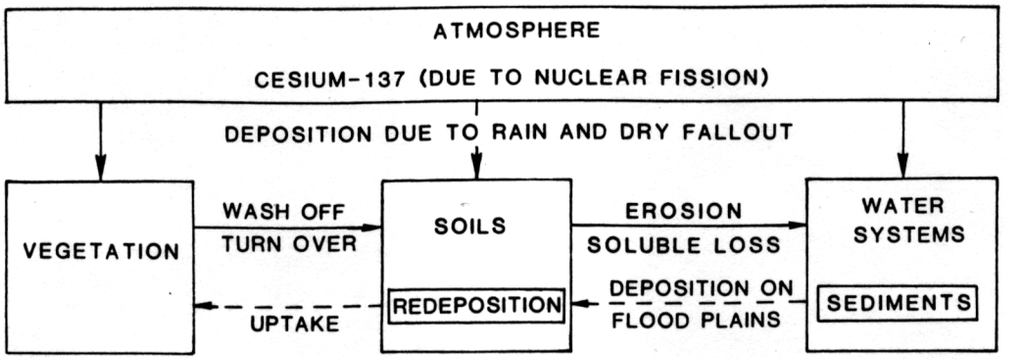

Fig. 2. Diagram of the ^{137}Cs cycle on the landscape.

1976). This suspended material is mostly eroded particles and becomes part of the sedimentation process. Except for overbank floods, which can redeposit ^{137}Cs on floodplains, deposited sediments in water bodies become the eventual sink for ^{137}Cs.

Although biological and chemical processes can move limited amounts of ^{137}Cs between vegetation, soil, and water, physical factors are the dominant processes transferring ^{137}Cs in natural systems (Davis, 1963; Rogowski and Tamura, 1970a,b; Ritchie et al., 1974; Dahlman et al., 1975). Because ^{137}Cs is uniformly distributed across the landscape and strongly adsorbed on soil particles, radioactive fallout ^{137}Cs can be used as a tracer for studying the physical processes of erosion and sedimentation. Thus, spatial measurements of ^{137}Cs can provide quantitative information on rates and patterns of erosion and sedimentation.

THE USE OF CESIUM-137 TO STUDY EROSION

Studies began in the early 1960s to measure the relationship between soil erosion and movement of fallout radionuclides. Menzel (1960) showed that loss of fallout ^{90}Sr from plots used to study soil erosion in Georgia and Wisconsin was related to measured soil loss. Graham (1963) found similar patterns for the movement of fallout ^{85}S and ^{131}I from soil erosion plots in Missouri. Frere and Roberts (1963) measured fallout ^{90}Sr loss from small research watersheds at Coshocton, OH, and found no loss of ^{90}Sr from grass-covered watersheds, whereas concentrations of ^{90}Sr in cultivated watersheds was only one-third to two-thirds that of the grass watersheds. They related this ^{90}Sr loss to measured soil loss from these watersheds. Dahlman and Auerbach (1968) found similar patterns for loss of ^{137}Cs applied to grass in Tennessee. Rogowski and Tamura (1965, 1970a,b) applied ^{137}Cs to small grassed test plots in Tennessee and found a logarithmic relationship between measured soil loss and ^{137}Cs loss. In 1974, a method was described for using measurements of the percentage loss of fallout ^{137}Cs to measure soil loss (Ritchie et al., 1974). In this article, fallout ^{137}Cs loss from different land uses in a watershed was shown to have a strong logarithmic relationship with soil loss estimated using the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978). Then they combined their results with those of Menzel (1960), Graham (1963), Frere and Roberts (1963), and Rogowski and Tamura (1970a,b) to show a strong logarithmic relationship (r = 0.94) between soil loss in metric tons per hectare per year and percent radionuclide loss (Fig. 3). They concluded that soil erosion rates can be reliably measured by determining the percent of fallout ^{137}Cs loss from soils.

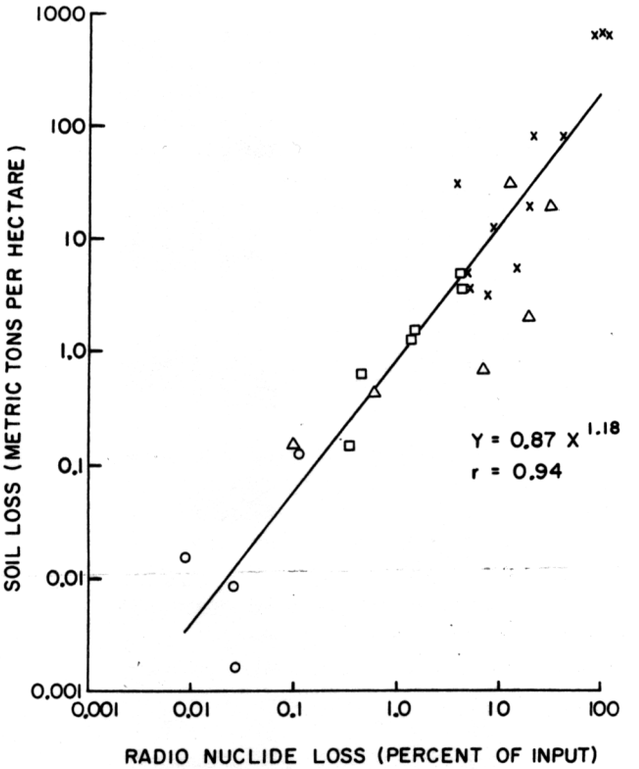

Fig. 3. Relationship between percent radionuclide loss and soil loss.

McHenry and Ritchie (Ritchie et al., 1971, 1974, 1982; McHenry and Ritchie, 1977a; McHenry et al., 1978; Mitchell et al., 1980; McHenry and Bubenzer, 1985; Spomer et al., 1985) began a series of studies in the early 1970s using the spatial distribution of ^{137}Cs in watersheds to determine (i) patterns of soil erosion and redeposition of eroded material within the watershed, (ii) movement of eroded material from watersheds, and (iii) deposition of eroded material in water bodies. These studies showed that ^{137}Cs measured at noneroded sites was equal to the fallout ^{137}Cs deposited in the area and was in the upper part of the soil profile or in the plow layer of cultivated soils and thus

could be used as the baseline for the input of ^{137}Cs to an area. At sites where soil erosion was occurring, total ^{137}Cs in the soil profile was less than the input from fallout. At deposition sites in the watershed, total ^{137}Cs was greater than input from fallout and/or the depth distribution of ^{137}Cs in soil profiles was greater than that measured in a noneroded site or deeper than the plow layer. McHenry and Ritchie (1977a) concluded that by measuring spatial patterns of ^{137}Cs in both vertical and horizontal planes across the landscape, rates of soil loss or deposition could be measured for different parts of a watershed. Even within landscape units, redistribution of soil could be mapped and erosion or deposition rates for different parts of individual fields measured and mapped from measurements of the spatial distribution of fallout ^{137}Cs. Erosion and deposition rates measured using the spatial distribution of ^{137}Cs were related to slope steepness, shape, and length. Flat areas at the top of slopes usually showed little soil loss. Flat areas at the base of slopes and concave slopes in fields often showed deposition. Thus measurements of the spatial distribution of ^{137}Cs could be used to show erosion patterns in fields, the redistribution of soils within fields, and soil loss from fields.

Martz and DeJong (1987) emphasized that the ^{137}Cs technique measures NET soil loss at each site rather than the total soil loss estimated by other methods and thus provides better information on true erosion rates and sediment delivery. Net soil loss was defined as soil loss minus soil deposition at a site. Other researchers have used ^{137}Cs in a wide variety of landscape and environmental conditions to measure soil loss, soil redeposition, and to develop budgets for erosion and sedimentation (i.e., Wise, 1977, 1980; McCallan et al., 1980; Brown et al., 1981a,b; Loughran et al., 1981, 1982, 1986a,b,c, 1987a,b; Campbell et al., 1982, 1985, 1986a,b; DeJong et al., 1982, 1983, 1986; Longmore et al., 1983a,b; Elliott et al., 1984; Kachanoski and DeJong, 1984; Kiss et al., 1986; Lance et al., 1986a,b,c; Walling et al., 1986a,b; Berghe and Gulinck, 1987; Kachanoski, 1987; Martz and DeJong, 1987; McIntyre et al., 1987; Menzel et al., 1987; Pennock and DeJong, 1987; Lowrance et al., 1988).

To use fallout ^{137}Cs to measure erosion, a baseline input of ^{137}Cs must be determined. This baseline input to the environment is due to fallout from the atmospheric nuclear testing rather than natural sources (Davis, 1963; Wise, 1980) and its initial distribution across the landscape is uniform or can be related to measurable environmental gradients (i.e., precipitation). It is not critical to have direct measurements of the actual ^{137}Cs deposited in a watershed, because local input of ^{137}Cs to a landscape unit can be estimated by measuring ^{137}Cs in soil profiles where neither erosion nor soil deposition has occurred (Ritchie et al., 1971, 1974; Campbell, 1982; Walling et al., 1986a,b; Martz and DeJong, 1987). It is important to assume a uniform deposition pattern across the landscape unit being studied or to be able to estimate the distribution pattern across a large watershed based on an environmental gradient so that measurements made at noneroding sites can be used to determine fallout input to the study area.

Another technique for determining baseline measurements is to develop a monitoring program and measure actual changes in ^{137}Cs with time at each site (Kachanoski and DeJong, 1984; Kachanoski, 1987). Such a monitoring program allows a comparison of actual measurements rather than having to make assumptions about uniform fallout patterns. Kachanoski (1987) has shown high variability in the spatial distribution of ^{137}Cs on landscape units.

Comparisons between baseline and current measurements of the vertical and horizontal distribution of ^{137}Cs across the landscape will reflect the movement of ^{137}Cs adsorbed on soil particles within and between soils and water (Fig. 2) by the physical process of erosion. Movement of ^{137}Cs in soils by biological and chemical processes is small (Tamura, 1964a,b) in comparison with movement of ^{137}Cs by physical processes (i.e., water and wind) that transport soil particles across the landscape. Loss of ^{137}Cs from a site is strongly related to the amount of soil loss (Ritchie et al., 1974; DeJong et al., 1982; Martz and DeJong, 1987). Thus, the spatial distribution of ^{137}Cs will reflect the pattern of atmospheric fallout as it has been modified by patterns of erosion and deposition of soil particles. McHenry and Bubenzer (1985) measured the spatial distribution of ^{137}Cs in a Wisconsin watershed and estimated that 50 to 100 times more soil was redistributed within a field than was removed from the field by erosion.

Walling et al. (1986b) incorporated data on input, redistribution, and output of ^{137}Cs from a watershed to measure erosion and sedimentation on a watershed with a single technique. Such watershed budgets of fallout ^{137}Cs (Ritchie et al., 1971, 1974, 1982; Ritchie and McHenry, 1976; McHenry and Ritchie, 1975; Bazzoffi and Panicucci, 1983; DeJong et al., 1983; Walling et al., 1986b) and studies of the spatial distribution of ^{137}Cs provide a unique source of data for studying and understanding erosion and sedimentation.

THE USE OF CESIUM-137 TO STUDY SEDIMENT ACCUMULATION

In 1961, Ravera (1961) published a report on the buildup of fallout radioactivity in sediments of Lake Maggiore in Italy. This report included a vertical profile of the radioactive beta-ray activity in the sediment. Later, using this sediment profile, Ravera and Premazzi (1971) reported that "the sedimentation rate was evaluated by knowing the distribution in time of the fall-out in air."

Studies in the mid 1960s by Pickering et al. (1965, 1966); Carringan and Pickering (1967); Pickering (1969) at Oak Ridge, TN, on the Clinch River; and Nelson et al. (1966) at Hanford, WA, on the Columbia River, related the vertical distribution of total gamma activity, total beta activity, ^{137}Cs, and other radionuclides in sediments to the time history of radionuclide release from Oak Ridge or Hanford nuclear processing facilities. These studies were based on relatively high levels of radioactivity compared with levels of radioactivity in the environment due to radioactive fallout from nuclear tests.

In 1968, Schreiber (1968) and Schreiber et al. (1968) related the vertical distribution of total beta counts in marine sediments from the North Adriatic Sea to at-

mospheric radioactive fallout rates from nuclear testing. They postulated that the vertical distribution of total beta activity in sediments was related to yearly fallout rates and thus could be used to date horizons in sediment profiles. They attributed most of the radioactivity measured to fallout ^{144}Ce and ^{90}Sr. Independent measurements of sediment accumulation rates for these marine sediment profiles by other methods were not available for comparison with the rates measured from the beta radioactivity profiles.

In the early 1970s, published reports of independent studies in the USA (Ritchie et al., 1970c,d, 1973; Robbins and Edgington, 1972, 1975; McHenry et al., 1973), Italy (Ravera and Premazzi, 1971), Switzerland (Krishnaswami et al., 1971), England (Pennington et al., 1973), and Israel (Stiller and Assaf, 1973) concluded that the vertical distribution of fallout ^{137}Cs in sediments could be related to the time distribution of radioactive fallout ^{137}Cs deposition from the atmosphere and thus used to infer a geochronology for sediment profiles. These studies measured sediment accumulation rates and patterns by measuring the vertical distribution of ^{137}Cs in sediment profiles. Ritchie et al. (1973) and Ritchie and McHenry (1975a) compared their sediment accumulation rates (cm yr^{-1}) measured from ^{137}Cs in sediment profiles with sediment accumulation rates measured at the same site by standard sediment survey methods (Fig. 4) and showed that comparable sedimentation rates were measured by the different techniques.

To use ^{137}Cs to measure sediment accumulation rates and patterns, the temporal pattern of fallout from atmospheric nuclear testing or release from nuclear facilities must be known (Davis, 1963; Wise, 1980). A comparison of these temporal measurements of the input of atmospheric fallout ^{137}Cs to the environment with measurements of ^{137}Cs in sediment profiles can be used to date different horizons in sediment profiles. By knowing the depth of these different ^{137}Cs horizons, rates of sediment accumulation can be calculated.

Cesium-137 input to sediment profiles is from direct deposit at the site or from the movement of ^{137}Cs deposited on the upstream watershed by the physical processes that transport eroded particles. In water bodies, ^{137}Cs deposited on the water surface is adsorbed to suspended materials in the water (Armstrong and Gloyna, 1969; Eyman and Kevern, 1975; Francis and Brinkley, 1976). This suspended material, which is primarily eroded soil particles also carrying ^{137}Cs, will be deposited as part of the sediment deposition process. Most of the suspended material entering a water body is deposited rapidly; thus, the sequential buildup of deposited sediment will reflect the time distribution pattern of fallout ^{137}Cs.

Because ^{137}Cs is strongly adsorbed on soil particles in the surface soil layer (Beck, 1966) and eroded materials tend to come from the same area within a watershed each year (Sprayberry and Bowie, 1969), ^{137}Cs concentration on soil particles being eroded from a watershed and moved to depositional sites will reflect the temporal fallout of ^{137}Cs.

Dates for two sediment horizons (1954 and 1964) can be identified in most deposition sites in the northern hemisphere based on the measurement of ^{137}Cs concentration. These horizons may not be as definitive in the southern hemisphere due to lower rates of fallout (Longmore, 1982). Sediment horizons for 1958 can sometimes be identified. A 1971 sediment horizon has been identified in the southern hemisphere (Campbell, 1983) and tentatively identified in a few northern hemisphere sediment profiles (McHenry et al., 1984). The Chernobyl accident has labeled a sediment horizon in much of Europe (Buesseler et al., 1987). Thus, with several depositional horizons marked by ^{137}Cs, com-

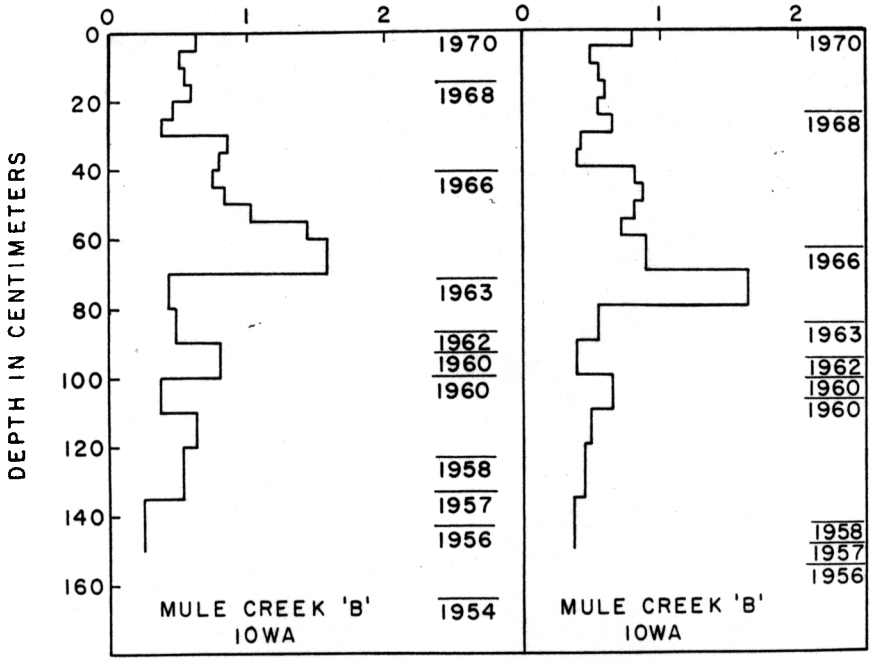

Fig. 4. Depth distribution of ^{137}Cs in sediment profiles from Mule Creek B reservoir in Iowa. Dates on the right of each profile indicate

parison of sediment accumulation rates can be made for different time periods from measurements of a single sediment profile.

The ^{137}Cs technique has some advantages over conventional techniques of measuring sediment accumulation rates. Measurements of sediment accumulation rates can be made rapidly from measurements of a single sediment profile with no requirement to return to the site to make repeated measurements. Interpretations about sediment accumulation rates are made by comparing the vertical distribution of ^{137}Cs in sediments with the known time deposition of fallout ^{137}Cs from the atmosphere. This time distribution (monthly and annual) of radioactive fallout has been extensively measured and documented for different areas around the world (Cambray et al., 1983; Larsen, 1985).

There are certain factors about the ^{137}Cs technique that must be considered when interpretations of the vertical distribution of ^{137}Cs in sediments are used to measure sediment accumulation rates. Only two dates (1954 and 1964) can be determined for most sediment profiles. If more detail on the time rate of sediment accumulation is needed, other techniques should be used.

Determining ^{137}Cs near the detection limits can cause inaccuracies in locating the 1954 sediment horizon. Radioactive decay of ^{137}Cs (half-life = 30 yr) continues to reduce the amount of ^{137}Cs in the environment and thus limits the detection of the 1954 horizon (Campbell, 1982) due to the small amount of ^{137}Cs originally present in the 1954 horizon. By 1984, only half of the original amount of ^{137}Cs deposited in the 1954 sediment horizon would be left. This is especially critical in the southern hemisphere where fallout rates are lower than in the northern hemisphere (Fig. 1).

Bulk density of the sediment sample must be known to be able to calculate total ^{137}Cs. Also, because ^{137}Cs is strongly adsorbed on clay-size particles, knowledge about the particle size distribution within the sediment profile is helpful in interpretation (Fig. 5). An increase

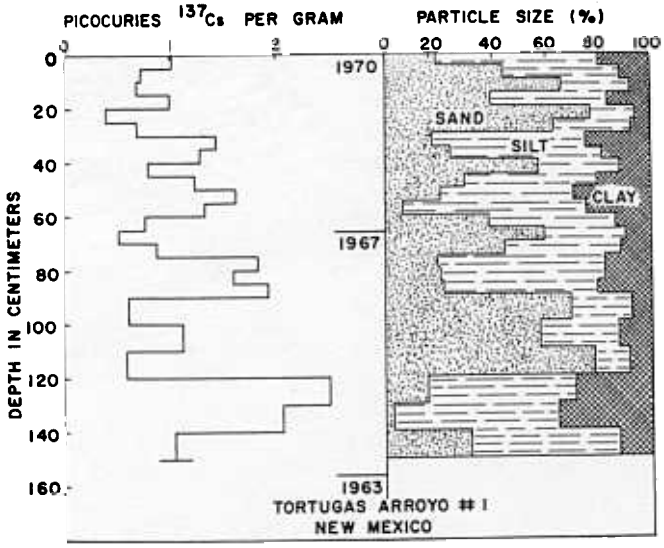

Fig. 5. Depth distribution of ^{137}Cs in a sediment profile from Tortugas Arroyo no. 1 reservoir in New Mexico compared with particle size distribution in the same profile.

in the amount of sand-size particles in sediment profiles will cause an apparent decrease in the concentration of ^{137}Cs at that point in the sediment profile that cannot be related to any time change in atmospheric fallout rates of ^{137}Cs. For some sediment profiles it is probably better to express ^{137}Cs concentration in units per unit of clay to account for the differences in ^{137}Cs distribution due to differences in particle size distribution in the sediment profile. In depositional environments (i.e., barrier islands) where sands dominate the sediment profile, ^{137}Cs will be of limited value in determining sedimentation.

A time lag exists between the time of atmospheric deposition of ^{137}Cs and the time of deposition of ^{137}Cs in sediment profiles. This time lag is from factors controlling the processes of sediment deposition. Early research in a small watershed indicated a time lag of 6 to 12 mo between the time of atmospheric fallout and the time of ^{137}Cs deposition in a reservoir sediment (Ritchie et al., 1973). However, the actual time lag will depend on the rates and frequency of different natural processes in each water body and watershed (Wise, 1980). Cesium-137 may even help understand these time lag processes (Walling et al., 1986a). This time lag should be considered very carefully in assigning dates to sediment horizons.

Physical reworking of sediments by animals, wave action, or other methods redistribute ^{137}Cs within the profile and thus spread ^{137}Cs over a larger part of the profile and reduce maximum concentrations of ^{137}Cs. A number of studies have reported this redistribution (i.e., Robbins and Edgington, 1975; Robbins et al., 1977b, 1979; Hakanson and Kallstrom, 1978; Krezoski et al., 1978; Krezoski and Robbins, 1985; Fisher et al., 1980; Kacieszczenko and Banasik, 1981; Demaster et al., 1985; Robbins, 1986a; Anderson et al., 1987; Sharma et al., 1987). Kacieszczenko and Banasik (1981) used a model to predict the effects of bioturbation on ^{137}Cs movement and showed that movement can occur. In general, any reworking of sediment broadens the ^{137}Cs peaks and makes interpretation more difficult. However, such movement will not likely change the position of the major ^{137}Cs horizons (Pennington et al., 1973, 1976; Ritchie et al., 1973; Krishnaswami et al., 1971; Robbins and Edgington, 1975; Wise, 1980; Campbell, 1982). Such movement, however, can have a major effect on the position of the ^{137}Cs horizon deposited in 1954. By spreading detectable levels of ^{137}Cs into lower parts of the profile, the first detectable level of ^{137}Cs would be lower than it should be. Localized reworking of the near-surface sediments can also displace ^{137}Cs horizons, especially in areas with low sediment deposition rates. In general, the lower the sediment accumulation rate the greater effect reworking of sediments will have on the redistribution of ^{137}Cs in sediment profiles.

Diffusional movement of ^{137}Cs also redistribute ^{137}Cs in sediment profiles. Although diffusional movement is usually limited (Davis, 1963; Tamura, 1964a,b; Aston and Duursma, 1973; Lomenick and Tamura, 1965), it does occur and can be a problem in certain environments (i.e., Krishnaswami et al., 1971; Evans et al., 1983; Davis et al., 1984; Torgenson and Longmore, 1984). Again, such movement will not likely change the position of ^{137}Cs horizons (Penning-

ton et al., 1973, 1976; Ritchie et al., 1973; Krishnaswami et al., 1971; Robbins and Edgington, 1975; Wise, 1980; Campbell, 1982). However, for soft-water lakes (Davis et al., 1984), diffusion of ^{137}Cs can be a major problem. Diffusional movement of ^{137}Cs has the greatest effect on the location of the ^{137}Cs horizon deposited in 1954; thus, care needs to be taken in determining the location of this horizon.

Cesium-137 is difficult to use in water bodies with low sediment accumulation rates (< 1 cm yr^{-1}) because of sampling problems. If sediment accumulation is low, then sampling of horizons in the detail necessary to determine ^{137}Cs horizons in the profile may be difficult.

The ^{137}Cs technique can be used only with sediments deposited since 1954, the year that radioactive fallout ^{137}Cs was first deposited in measurable amounts. Also, as noted earlier, determining the position of the 1954 horizon will become more difficult with time due to the natural decay of ^{137}Cs. Thus, one is limited to determining sediment accumulation rates since 1954. Other techniques (e.g., ^{210}Pb, ^{14}C, pollen) can be used to measure older deposits.

Fallout ^{137}Cs has been declining in the environment since 1964; thus, the last major sediment ^{137}Cs marker is usually 1964. For determinations of sediment accumulation rates after 1964 or sediment accumulation rates in a reservoir built since 1964, the ^{137}Cs technique may not be applicable. There are exceptions to this, such as a horizon labeled by fallout from the Chernobyl accident in Europe or other similar events.

SUMMARY

Research beginning in the late 1960s has shown that ^{137}Cs deposited across the landscape by fallout from nuclear test is a unique and useful tool for studying erosion and sedimentation. Cesium-137 has been used to measure erosion rate and delineate patterns of erosion and deposition within landscape units to provide data on net erosion from a site. These data provide a basis for determining budgets and patterns of soil redistribution within a field and actual soil loss from the field. Cesium-137 has also been used to determine the sediment accumulation rates in a wide variety of depositional environments for sediment deposited since 1954. By providing data on erosion rates and patterns and on sediment accumulation rates and patterns, ^{137}Cs is a unique tool for studying the complete erosion and sedimentation cycle across the landscape.

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The References section contains publications and reprints collected by the authors over the time that they have been involved in research on the use of fallout ^{137}Cs to study erosion and sedimentation. In our first paper on ^{137}Cs (Ritchie et al., 1970b), we stated that our purpose was "to describe the use of radioactive fallout, as ^{137}Cs, as a tracer of erosion, sediment transport, and sediment deposition within a watershed." Our research and that of our colleagues have more than met this purpose as stated at the meeting of the Mississippi Water Resources Conference on 15 Apr. 1970.

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